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FOCUS ARTICLE

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Progress in reducing nutrient and sediment loads to Chesapeake Bay: Three decades of monitoring data and implications for restoring complex ecosystems

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Abstract

For over three decades, Chesapeake Bay (USA) has been the focal point of a coordinated restoration strategy implemented through a partnership of governmental and nongovernmental entities, which has been a classical model for coastal restoration worldwide. This synthesis aims to provide resource managers and estuarine scientists with a clearer perspective of the magnitude of changes in water quality within the Bay watershed, including nitrogen (N), phosphorus (P), and sediment for the River Input Monitoring (RIM) watershed and the unmonitored below-RIM watershed. The flow-normalized N load from the RIM watershed has declined in the period of 1985–2017, but P and sediment loads have lacked progress. Reductions of riverine N are largely driven by reductions of point sources and atmospheric deposition. Future reductions will require significant progress in managing agricultural nonpoint sources. The below-RIM watershed, which comprises a disproportionately high fraction of inputs to the Bay, has shown long-term declines in major sources, including point sources (N and P), atmospheric deposition (N), manure (N and P) and fertilizer (P), based on a combination of monitoring and modeling assessments. To date, the Bay cleanup efforts have achieved some progress toward reducing nutrients from the watershed, which have resulted in improving water quality in the estuary. However, further reductions are critical to achieve the Chesapeake Bay Total Maximum Daily Load goals, and emerging challenges due to Conowingo Reservoir, legacy nutrients, climate change, and population growth should be considered. Continued monitoring, modeling, and assessment are critically important for informing the restoration of this complex ecosystem.

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Chesapeake Bay, nonpoint source, point source, restoration, trend analysis, water quality

1 | TOWARD RESTORING A COMPLEX ECOSYSTEM

Coastal environments have been adversely impacted by anthropogenic riverine inputs of nutrients and sediment, resulting in eutrophication and hypoxia, declines in water transparency, and loss of submerged aquatic vegetation (Boesch, 2019; Breitburg et al., 2018). One notable example is Chesapeake Bay, which is among the largest estuaries in North America (Boesch et al., 2001; Kemp et al., 2005). Formed about 10,000 years ago, Chesapeake Bay is home to many species of finfish, shellfish, plant, and underwater grasses. Its watershed spans ∼165,760 km^2, covering parts of 6 states, namely, Delaware, Maryland, New York, Pennsylvania, Virginia, and West Virginia, and the entire District of Columbia (Chesapeake Bay Program, 2022; Figure 1). Among the Bay's many tributaries, Susquehanna River is the largest (Chesapeake Bay Program, 2022).

Chesapeake Bay has a land-to-water ratio of 14:1, which is the largest of any coastal waterbody in the world (Chesapeake Bay Program, 2022). Therefore, the Bay is particularly vulnerable to human actions in the watershed. Since 1950, human population has more than doubled to about 18 million in 2017 and will reach 20 million by 2030 (Chesapeake Bay Program, 2020). Concurrently, profound changes have occurred in the watershed, including the expansion of urbanization and associated wastewater and stormwater runoff, increased air emissions from mobile sources, and enhanced application of fertilizer for crop production (Lyerly et al., 2014). Consequently, excessive nutrients and sediment have been delivered to the Bay from multiple major sources (e.g., wastewater, agricultural fertilizer and manure, atmospheric deposition; Ator et al., 2020; Ator et al., 2011; Sabo et al., 2022), resulting in ecological degradation in the estuary (Boesch, 2019; Boesch et al., 2001; Cerco & Noel, 2013; Harding et al., 2019, 2020; Kemp et al., 2005).

To protect Chesapeake Bay and its living resources, restoration efforts have been pursued to reduce nutrient and sediment loads as outlined in a series of Chesapeake Bay Partnership Agreements (Chesapeake Executive Council, 1987, 1992, 2000, 2014; Chesapeake Bay Partnership, 1983) and the Chesapeake Bay Total Maximum Daily Load (TMDL; U.S. Environmental Protection Agency [EPA], 2010). The 1983 Partnership Agreement, signed by the EPA with Maryland, Pennsylvania, Virginia, and the District of Columbia, was the first to recognize the need for cleanup actions. This also led to the formation of the Chesapeake Bay Program for the purpose of regional restoration. The 1987 Agreement set a goal to reduce nutrient loads to the Bay by 40% by 2000. The 2000 Agreement set a goal of improving the water quality of the Bay to delist it from EPA's list of "impaired waters" by 2010. Although these agreements promoted coordination among the Bay jurisdictions, restoration efforts were largely voluntary, which turned out to be ineffective (EPA, 2010). Consequently, the TMDL was established in 2010 to set limits on the amount of total nitrogen (TN), total phosphorus (TP), and suspended sediment (SS) entering the Bay to meet water quality conditions that can fully support living resource survival, growth, and reproduction (Keisman & Shenk, 2013; Tango & Batiuk, 2013). Under the TMDL, the seven jurisdictions independently developed Watershed Implementation Plans aimed toward fully meeting the goals by 2025 (Linker et al., 2013; Shenk & Linker, 2013).

Long-term water-quality monitoring of the Bay watershed is an essential component of the restoration effort. The Bay Program Partnership initiated the River Input Monitoring (RIM) network in 1985 to monitor water quantity and quality at nine stations near the fall-line, or upstream from tidal influence, of nine major tributaries (Figure 1 and Table S1), which was later expanded to a nontidal monitoring network (NTN) of over 100 stations (Langland et al., 2012). Since 2012, the US Geological Survey (USGS) has adopted the Weighted Regressions on Time, Discharge, and Season (WRTDS) method to quantify nutrient and sediment loads and trends at the monitoring stations (Hirsch et al., 2010). Using a statistical smoothing procedure called "flow-normalization," WRTDS can effectively remove interannual variability in streamflow and better detect trends in loads. Thus, it has been used extensively to compute and

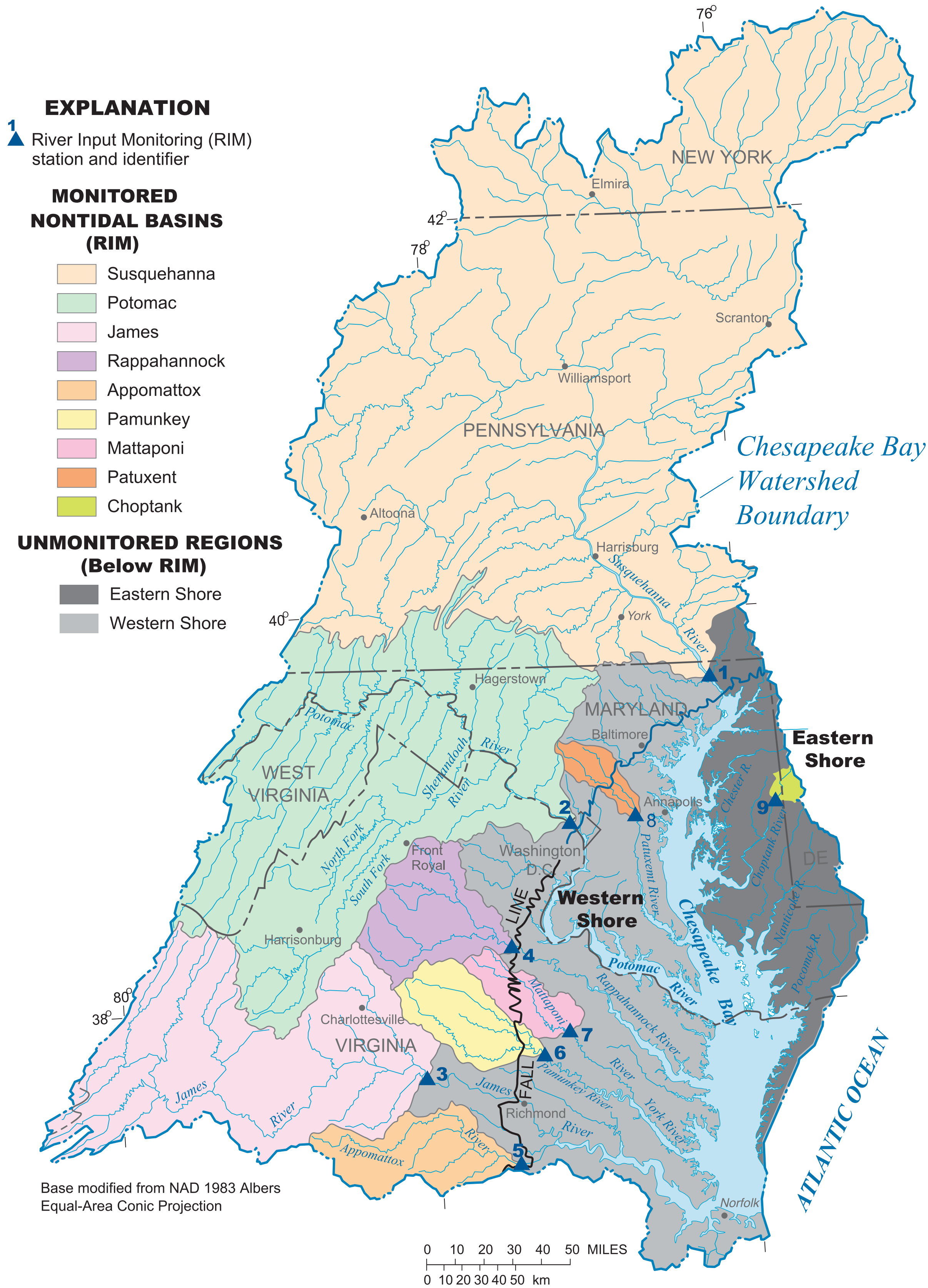

FIGURE 1 The Chesapeake Bay watershed and the nine River Input Monitoring (RIM) stations. These stations are located at the fall-line, or upstream from tidal influence, of nine major tributaries, namely, Susquehanna River (1), Potomac River (2), James River (3), Rappahannock River (4), Appomattox River (5), Pamunkey River (6), Mattaponi River (7), Patuxent River (8), and Choptank River (9). More details are presented in Table S1.

communicate flow-normalized (FN) long-term trends at stations in the RIM network (Hirsch et al., 2010; Moyer et al., 2012; Moyer & Blomquist, 2018; Zhang et al., 2015) and the NTN (Chanat et al., 2016; Moyer et al., 2017).

Water-quality and ecological conditions in Chesapeake Bay, however, are responsive to the actual amounts of nutrients and sediment received in the preceding days, weeks, or months rather than the FN loads. In this regard, WRTDS also calculates actual (or "true-condition") loads, which correspond to the observed streamflow conditions. However, to our knowledge, long-term trends in WRTDS true-condition loads (or concentrations) have rarely been reported for the RIM tributaries. Another major knowledge gap is the export of nutrient and sediment from the watersheds downstream of the RIM stations (hereafter, "below-RIM watershed"; Figure 1), including point sources, land-based nonpoint sources, shoreline erosion, and atmospheric deposition to the surface of the tidal waters. The RIM and below-RIM watersheds roughly represent the monitored nontidal and downstream unmonitored parts of the Bay watershed, respectively. Data for the latter were obtained from the Chesapeake Bay Watershed Model (Chesapeake Bay Program, 2017).

The main objective of this synthesis is to provide resource managers and estuarine scientists with a clearer perspective on a key management question: After three decades of restoration, has progress been made in reducing nitrogen, phosphorus, and sediment exports from the Bay watershed? This synthesis leverages available monitoring, modeling, and research to quantify long-term trends in the true-condition RIM tributary loads as well as the various major sources from the below-RIM watershed. In addition, we provide a synthesis of literature on estuarine response to changes in nutrient loads and emerging challenges to load reductions. We conclude with reflections and insights on the Bay restoration efforts in reference to this synthesis.

2 | APPROACHES TO ASSESS PROGRESS IN REDUCING NUTRIENTS AND SEDIMENT

2.1 | RIM watershed

WRTDS true-condition and FN estimates of TN, TP, and SS loads and concentrations have been reported by the USGS for the RIM stations for the period of 1985–2017 (Moyer & Blomquist, 2018). By accounting for interannual streamflow variability, FN estimates can better reveal trends and detect impacts of management strategies (Hirsch et al., 2010). The FN trends fall into three categories based on likelihood values obtained from the WRTDS Bootstrap Test (Hirsch et al., 2015): (1) "Declining" indicates a negative change in load with a likelihood ≥0.67; (2) "Increasing" indicates a positive change with a likelihood ≥0.67; and (3) "No Trend" indicates any change with a likelihood <0.67. In addition, the annual total inputs from the nine RIM stations (called "RIM total" or "RIM") were quantified for the FN loads. Unlike the individual tributaries, the trend for the RIM total FN load cannot be directly computed due to the lack of approach; instead, it was inferred from the trend direction and significance of the three largest tributaries, which account for over 90% of RIM total FN load.

As noted above, WRTDS true-condition estimates represent the actual amounts of RIM tributary input that enter the tidal estuary and are directly relevant to estuarine conditions and changes. Thus, long-term (1985–2017) trends in true-condition estimates were quantified for each RIM station as well as the RIM total. Specifically, the Mann–Kendall test and Sen's slope were applied to true-condition annual loads, annual yields (i.e., annual load / watershed area), and flow-weighted concentrations ("FWC"; i.e., annual load/annual river flow), respectively. The Mann–Kendall test is a nonparametric approach for detecting monotonic trends (Kendall, 1975) and Sen's slope quantifies trend magnitudes (Sen, 1968). These approaches were implemented through the "mannKen" function in the "wq" package (Jassby & Cloern, 2016) using the R software (R Core Team, 2019). Like the FN trends, true-condition trends fall into three categories: "Declining" indicates a negative slope in load, yield, or concentration with a p-value <0.1; (2) "Increasing" indicates a positive slope with a p-value <0.1; and (3) "No Trend" indicates any slope with a p-value ≥0.1.

2.2 | Below-RIM watershed

The below-RIM areas are largely unmonitored (Figure 1). However, major sources of nutrient and sediment can be obtained from the Bay Program Partnership's Phase 6 Watershed Model, which provides annual time series of point (wastewaters), atmospheric, nonpoint, and shoreline sources (Chesapeake Bay Program, 2017). Long-term average annual loads for the period of 1985–2014 were calculated for each below-RIM source as well as the RIM tributaries to assess the relative contribution of each component to the total input to the Bay. In general, point and atmospheric sources are more certain because they were derived from observations. By contrast, nonpoint and shoreline sources are more uncertain because they were estimated based on the watershed model calibrated with limited monitoring data in the below-RIM areas. Thus, they were only used for quantifying the relative contribution of each component to the total input to the Bay (Section 3.1). For long-term trends (Section 3.2), major sources of N and P for the below-RIM watershed, including point sources, atmospheric deposition directly to the tidal waters, and nonpoint sources (i.e., fertilizer and manure applied to the land), were analyzed for the period of 1985–2017 using the Mann-Kendall test and Sen's slope. For completeness, this analysis was also conducted for the RIM watershed.

3 | PROGRESS IN REDUCING NUTRIENTS AND SEDIMENT LOADS

3.1 | Long-term average loads (1985–2014)

RIM tributaries yielded the majority of river flow (Q) (78%), TN (62%), TP (60%), and SS (44%) loads to the Bay (Figure 2 and Table S2). Collectively, the nine tributaries contributed 56 km^3 of Q, 92,000 Mg of TN, 6000 Mg of TP, and 4,200,000 Mg of SS to the Bay annually during the period of 1985–2014. Among these tributaries, watershed area, land use, and physiography vary considerably (Table S1), resulting in different relative contributions to the RIM total load. Susquehanna River, the largest tributary to the Bay, accounted for 63% of Q, 66% of TN, 45% of TP, and 45% of SS of the RIM total inputs. The lower contributions of TP and SS by Susquehanna than Q and TN reflect historical retention of TP and SS by the reservoirs system above Conowingo Dam (Langland, 2009; Langland & Hainly, 1997). Potomac and James Rivers are the second and third largest contributors. Together, these three tributaries accounted for 92% of Q, 95% of TN, 91% of TP, and 92% of SS of the RIM total inputs.

The below-RIM areas contributed substantial, additional amounts of Q, TN, TP, and SS to the Bay (Figure 2 and Table S2). For the period of 1985–2014, point sources from the below-RIM areas contributed 22,000 Mg of TN, 1300 Mg of TP, and 20,000 Mg of SS to the Bay, annually. Moreover, land-based nonpoint sources from the below-RIM areas provided even larger inputs than point sources. Furthermore, shoreline erosion delivered substantial amounts of TP and SS to the Bay. Together, these three sources delivered 57,000 Mg of TN, 3900 Mg of TP, and 5,300,000 Mg of SS to the Bay, annually. These estimates indicate that both the monitored RIM watershed and the unmonitored below-RIM watershed are important contributors of nutrient and sediment loads to the Bay.

3.2 | Long-term trends in loads (1985–2017)

FN loads are used to quantify long-term trends in nutrient and suspended sediment loads at the RIM stations for the period of 1985–2017 (Figures 3 and 4 and Table 1). For the RIM total input, TN load showed a long-term decline (−19%) over the period; however, TP and SS loads showed minimal changes at −2.5% and −1.5%, respectively (Figure 3a–c and Table 1). Although there is no approach for directly computing trend significance for the RIM total load, the estimated decline in TN is likely statistically significant because seven major tributaries have had declining trends, including the five largest tributaries. By contrast, the estimated declines in TP and SS are small, which are likely statistically nonsignificant because the major tributaries have had mixed trends. Trends vary among the nine RIM tributaries (Figure 4a–c and Table 1). For TN, seven tributaries had declining trends and two had increasing trends (i.e., Pamunkey and Choptank). For TP, three had declining trends (i.e., Potomac, James, and Patuxent), five had increasing trends (e.g., Susquehanna), and two showed no trend. For SS, three had declining trends (i.e., Potomac, Patuxent, and Choptank), four had increasing trends (e.g., Susquehanna, James, Rappahannock, and Pamunkey), and two showed no trend. Considering both TN and TP, only Potomac, James, and Patuxent had declining trends, whereas Pamunkey and Choptank had increasing trends. Considering all three constituents, only Potomac and Patuxent had declining trends, whereas Pamunkey had increasing trends.

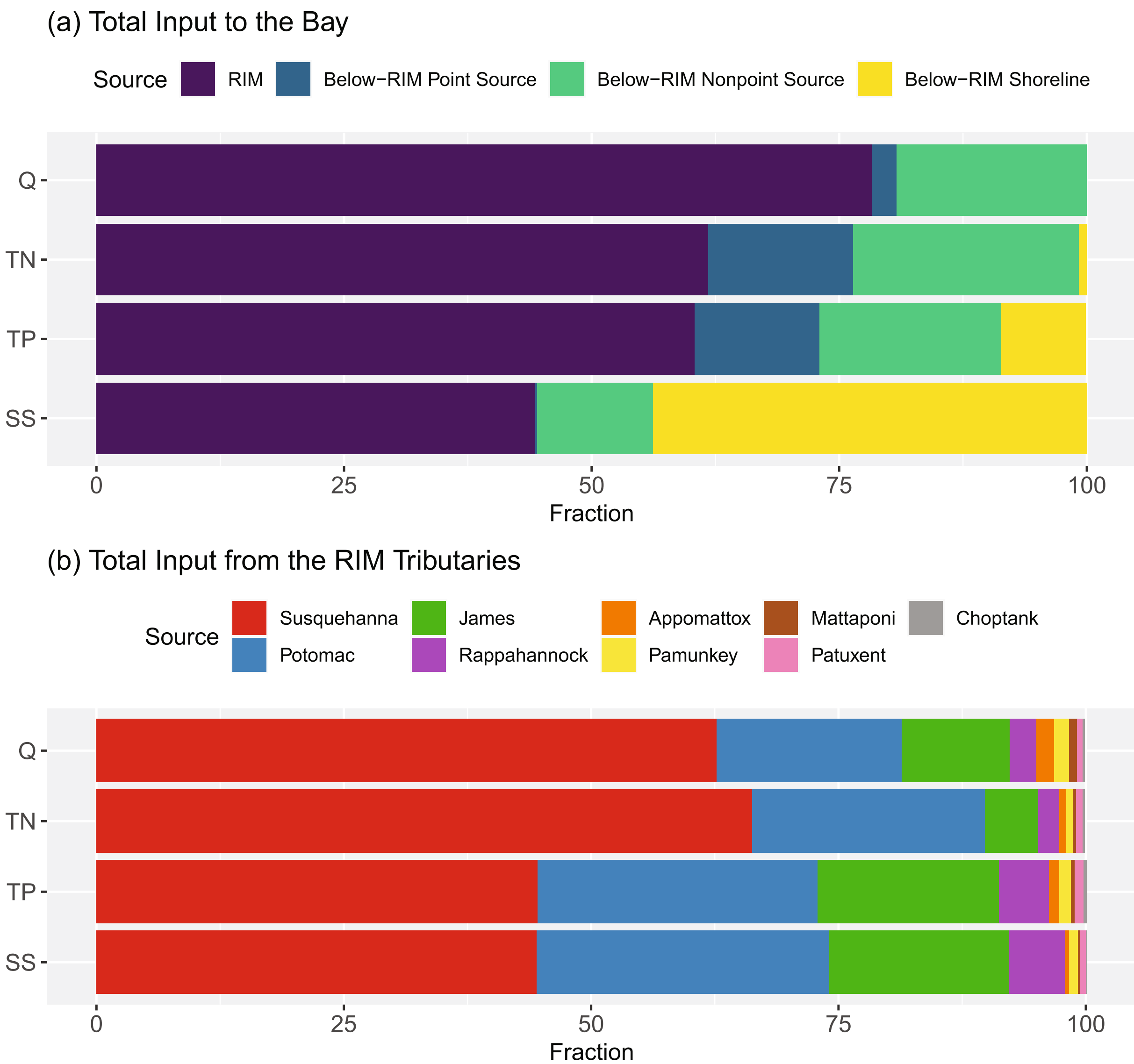

FIGURE 2 Relative contributions of river flow (*Q*), total nitrogen (TN), total phosphorus (TP), and suspended sediment (SS) in the concurrent period of 1985–2014: (a) total input to the Bay and (b) total input from the River Input Monitoring (RIM) watershed. More details are presented in Table S2. Data for the RIM watershed are from the US Geological Survey (USGS; Moyer & Blomquist, 2018); data for the below-RIM watershed are from the Chesapeake Bay Phase 6 Watershed Model (Chesapeake Bay Program, 2017).

True-condition loads represent the actual amounts of input to the Bay and are directly relevant to the conditions and changes observed in the estuary amidst the variability caused by weather (Figures 3d–f and 4d–f and Table 1). Because the true-condition loads are more variable than FN loads, their trend estimates had fewer significant trends than FN loads (Table 1). For the RIM total input, true-condition load does not show a significant trend for any of the three constituents (Figure 3d–f and Table 1). Similarly, most of the tributaries showed no long-term trend (Figure 4d–f and Table 1). Statistically significant trends only were observed in the James (declining TN and TP), Patuxent (declining TN), and Choptank (increasing TN and TP). These five cases are consistent with the FN trend directions.

True-condition FWC is a measure that also reduces the effects of interannual variability in streamflow. Correspondingly, FWC estimates had more occurrences of significant trends than true-condition loads but fewer than FN loads (Figures 3g–i and 4g–i and Table 1). For the RIM total input, FWC showed a long-term decline in TN but no trend in TP or SS (Figure 3g–i and Table 1), which is consistent with the FN trends. Trends vary among the nine RIM tributaries (Figure 4g–i and Table 1). For TN FWC, four tributaries had declining trends (i.e., the three largest tributaries and Patuxent), two had increasing trends (i.e., Pamunkey and Choptank), and three showed no change. For TP

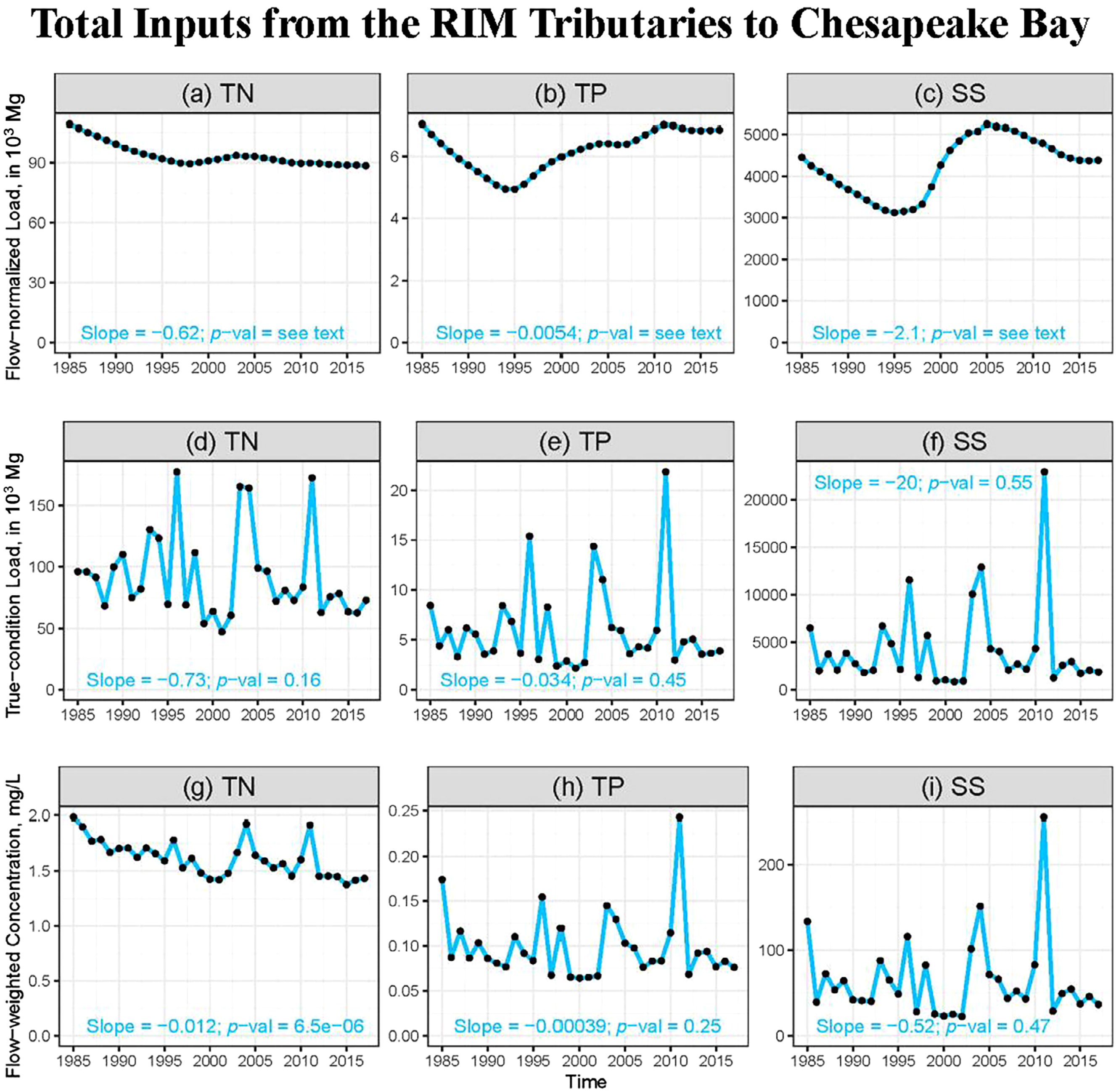

FIGURE 3 Total nitrogen (TN), total phosphorus (TP), and suspended sediment (SS) entering Chesapeake Bay from the River Input Monitoring (RIM) tributaries annually in the period of 1985–2017, as quantified using flow-normalized annual loads (top panel; a–c), true-condition annual loads (middle panel; d–f), and true-condition, flow-weighted concentrations (bottom panel; g–i). For the flow-normalized loads (a–c), trend significance cannot be directly computed due to the lack of approach – see the text and Table 1 for more details.

FWC, two had declining trends (i.e., James and Patuxent), four had increasing trends (i.e., Rappahannock, Appomattox, Pamunkey, and Choptank), and three showed no trend. For SS FWC, only one had an increasing trend (i.e., Pamunkey), and all others showed no trend. As in the case of true-condition loads, these significant trends identified in the FWCs are all consistent with the FN trends directions.

Overall, long-term trends in true-condition loads are somewhat inconsistent with those in FN loads. These differences are expected and should be clearly noted to end users when communicating the trends. Specifically, FN loads are more helpful for understanding changes in the upstream watershed that are related to management actions. However, for understanding the effects of riverine inputs on estuarine conditions and changes, true-condition estimates are more appropriate as they represent the actual inputs entering the Bay. Moreover, we note that while trends in true-condition loads are masked by interannual variability in streamflow, FWC can be a useful, alternative measure for detecting trends in the actual quality of freshwater entering the Bay.

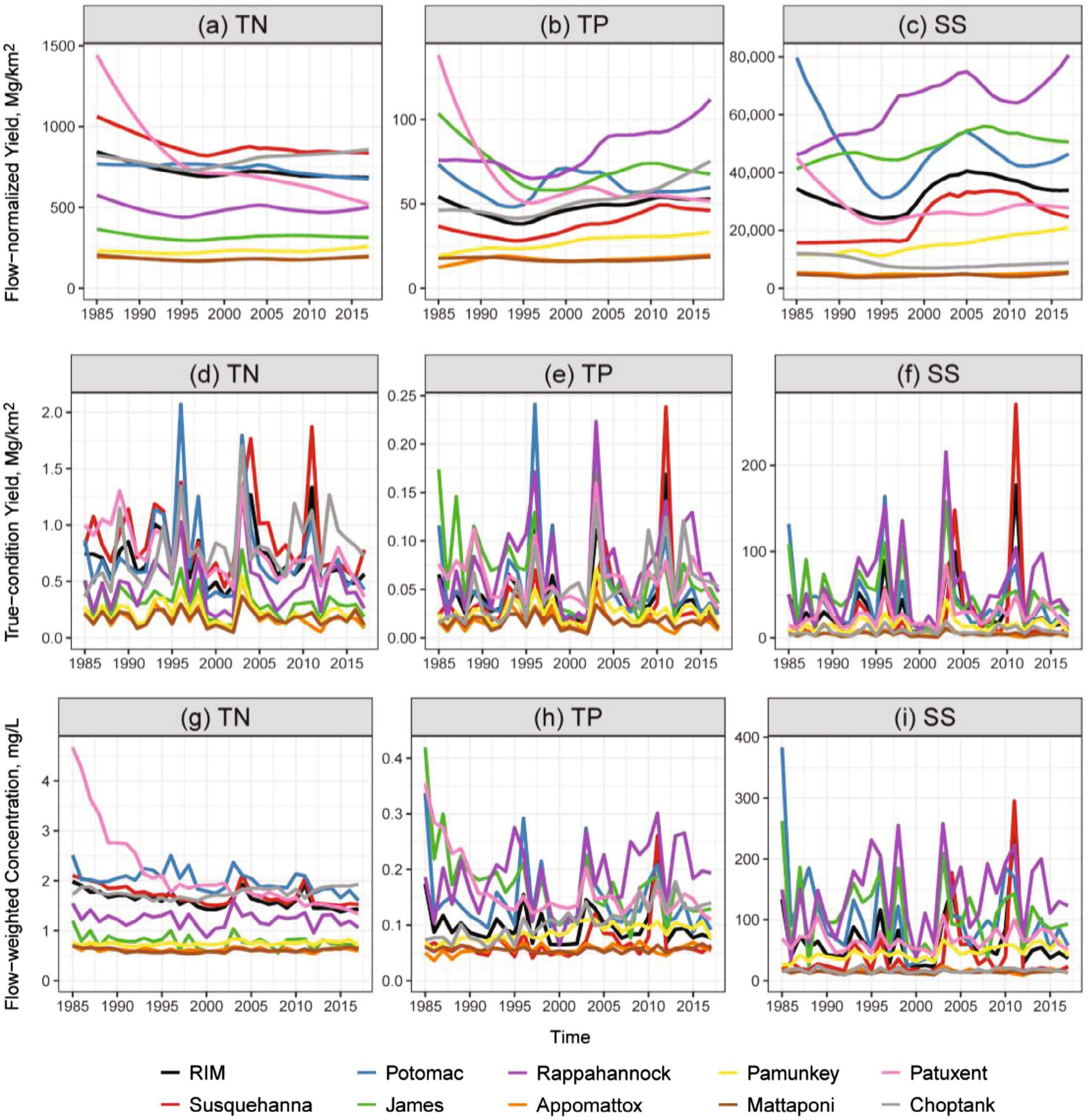

FIGURE 4 Total nitrogen (TN), total phosphorus (TP), and suspended sediment (SS) entering Chesapeake Bay from the River Input Monitoring (RIM) tributaries annually in the period of 1985–2017, as quantified using flow-normalized annual yields (top panel; a–c), true-condition annual yields (middle panel; d–f), and true-condition, flow-weighted concentration (bottom panel; g–i).

4 | PROGRESS IN REDUCING NUTRIENTS FROM MAJOR SOURCES

4.1 | Major progress in reducing N and P from point sources

A demonstrated success of the Bay restoration efforts is the reduction of nutrient loads from point sources (Table 2 and Figures S1 and S2). Point sources are generally more convenient to track and control than nonpoint sources because of their confined and identifiable locations. Over the last three decades, many industrial and municipal wastewater

TABLE 1 Long-term (1985–2017) trends (in percent) of total nitrogen (TN), total phosphorus (TP), and suspended sediment (SS) from the River Input Monitoring (RIM) tributaries.

	Flow-normalized load			True-condition load (or yield)			Flow-weighted concentration		
Rivers	**TN**	**TP**	**SS**	**TN**	**TP**	**SS**	**TN**	**TP**	**SS**
RIM total	−19[a]	−2.5[b]	−1.5[b]	−25	−13	−10	−19	−7.4	−13
Tributaries									
Susquehanna	−21	26	57	−23	13	0.55	−22	8.2	−4.0
Potomac	−12	−18	−42	−17	−9.3	−11	−15	−7.2	−14
James	−14	−35	23	−27	−29	−24	−13	−23	−9.6
Rappahannock	−13	48	75	0.33	33	38	−7.4	37	30
Appomattox	−0.63	59	4.7	−24	−16	−18	−2.3	17	−2.1
Pamunkey	11	73	79	−14	19	37	6.8	58	100
Mattaponi	−3.6	4.9	8.4	−6.4	−3.0	−6.7	2.2	−2.9	1.0
Patuxent	−64	−63	−38	−49	−24	68	−33	−20	9.5
Choptank	4.0	63	−27	106	297	102	11	107	7.9
Trend count[c]									
Declining	7	3	3	2	1	0	4	2	0
Increasing	2	5	4	1	1	0	2	4	1
No Trend	0	1	2	6	7	9	3	3	8

Note: Green cells indicate declining trends; yellow cells indicate increasing trends; and gray cells indicate no trend.
[a]No color given due to the lack of approach for directly computing trend significance for the RIM total load. However, the estimated decline in TN load is large, which is likely statistically significant because seven major tributaries have had declining trends, including the five largest tributaries.
[b]No color given due to the lack of approach for directly computing trend significance for the RIM total load. However, the estimated declines in TP and SS are small, which are likely statistically non-significant because the major tributaries have had mixed trends.
[c]Count of trend categories is only for the individual tributaries ($n = 9$).

TABLE 2 Long-term (1985–2017) trends of total nitrogen (TN) and total phosphorus (TP) from major sources to the River Input Monitoring (RIM) watershed and the below-RIM watershed.

		RIM watershed			Below-RIM watershed		
Source		**Slope, 10^3 Mg/year**	**Slope, Mg/km^2/year**	***p*-value[a]**	**Slope, 10^3 Mg/year**	**Slope, Mg/km^2/year**	***p*-value[a]**
Point Source	TN	−0.15	−1.1	***	−0.72	−21	***
	TP	−0.080	−0.62	***	−0.044	−1.3	***
Fertilizer	TN	−0.10	−0.80	—	0.44	13	***
	TP	−0.64	−4.9	***	−0.19	−5.4	***
Manure	TN	0.091	0.70	*	−0.12	−3.4	***
	TP	0.028	0.21	—	−0.056	−1.6	***
Atmospheric deposition	TN	−2.4	−19	***	−0.56	−16	***

Note: Green cells indicate declining trends; yellow cells indicate increasing trends; gray cells indicate no trend.
[a]Slope significance: *** $p < 0.01$, ** $p < 0.05$, * $p < 0.1$; — $p > 0.1$.
Source: Chesapeake Bay Program (2017).

treatment facilities across the Bay watershed have reduced nutrient effluent concentrations, because of more stringent regulations and improved treatment technologies (Ator et al., 2020; Chesapeake Bay Program, 2004, 2017).

The four largest RIM tributaries (i.e., Susquehanna, Potomac, James, and Rappahannock) and the Patuxent all had statistically significant, long-term declines in TN loads from point sources, whereas only Mattaponi had a statistically significant, long-term increase (Figure S1). The five largest RIM tributaries (i.e., Susquehanna, Potomac, James, Rappahannock, and Appomattox) and the Patuxent all had statistically significant, long-term declines in TP loads from point sources, whereas only Mattaponi and Choptank had statistically significant, long-term increases (Figure S2). Among these tributaries, Patuxent is a unique example where point source controls have resulted in water-quality improvements (Boynton et al., 2008), since it has a larger proportion of urban land and a larger contribution of point sources (Ator et al., 2011). Point source controls in this watershed included a statewide ban on P detergent in 1985 and implementation of biological nutrient removal (BNR) in the early 1990s (Boynton et al., 2008). Compared with the Patuxent, the Susquehanna, Potomac, and James rivers showed similar timing in TP reduction, likely reflecting similar timing of P-detergent bans in Maryland, Virginia, and Pennsylvania (Litke, 1999). However, the timing of TN reduction was more dissimilar, suggesting different timings for adopting advanced nutrient removal technologies.

Overall, point sources declined in both the RIM and the below-RIM watersheds during the period of 1985–2017 (Table 2). For TN, point sources from the RIM watershed had an estimated slope of −150 Mg/year ($p < 0.01$), whereas point sources from the below-RIM watershed had a much stronger decline (slope = −720 Mg/year; $p < 0.01$). For TP, point sources from the RIM watershed had a slope of −80 Mg/year ($p < 0.01$), while point sources from the below-RIM watershed had a smaller reduction (slope = −44 Mg/year; $p < 0.01$). These point source reductions in the below-RIM watershed have been linked to water-quality changes in the tidal waters. For example, the Blue Plains Advanced Wastewater Treatment Plant on the Potomac River, which is also the largest point source facility in the Bay watershed, had tremendous successes in reducing nutrients due to P-detergent bans in Maryland (1985) and Virginia and Washington, DC (1986) and the implementation of partial BNR (1996) and later full BNR (2000) (Lyerly et al., 2014; Ruhl & Rybicki, 2010). In the tidal Choptank, most wastewater treatment plants were upgraded to advanced treatment after 2000, resulting in significant reductions of TN and TP (Fisher et al., 2021).

Additional reductions of N and P from point sources may be achieved by investment in advanced treatment technologies at additional facilities across the Bay watershed. In this regard, facilities located closer to the tidal waters may lead to disproportionally large benefits on improving tidal water quality due to high delivery rates (i.e., less attenuation) along the river corridors. However, there is an upper limit in the total reductions achievable through point sources. It is not adequate to manage just point sources because they represent only 16% of TN and 32% of TP inputs from the Bay watershed (Ator et al., 2011).

4.2 | Additional reductions of N achieved by controlling atmospheric deposition

Another success of nutrient reduction is through the control of atmospheric N deposition (Table 2 and Figure S3). Although the 1990 Clean Air Act Amendments and related regulations on mobile and stationary sources were intended for improving air quality, co-benefits have been achieved toward improving river water quality (Eshleman et al., 2013; Eshleman & Sabo, 2016). During the period of 1985–2017, atmospheric deposition of N to the land surface showed statistically significant declines in all nine RIM watersheds (Figure S3). The temporal trajectory is quite similar among these watersheds, showing an overall decline since 1990, consistent with the promulgation of the Clean Air Act Amendments. The trend magnitudes were also quite comparable among these watersheds, ranging between −0.024 and −0.015 Mg/km^2/year. One exception is the Choptank watershed, which had a smaller decline (−0.009 Mg/km^2/year), likely reflecting the countereffect of increasing N emissions from animal farms on the Delmarva Peninsula (Ator & Denver, 2015; Keisman et al., 2018). Similar increases in N emissions from animal operations were suggested for the agriculturally intensive Lower Susquehanna watershed (Zhang, Ball, & Moyer, 2016).

Atmospheric deposition of N has declined in both the RIM and the below-RIM watersheds during the period of 1985–2017. The RIM watershed had a decline of −2400 Mg/year ($p < 0.01$), whereas the below-RIM watershed had a smaller decline of −560 Mg/year ($p < 0.01$) due to its smaller watershed size. When normalized by watershed size, the trend slope is similar between the RIM and below-RIM watersheds (Table 2).

Additional reductions of N from atmospheric deposition may be achieved by setting more stringent standards on automobile emissions, encouraging public transportation, reducing outputs of electric generating units, and/or reducing ammonia emissions from livestock waste through altered animal feeding (Lyerly et al., 2014). However, these measures may be impeded by projected population growth and urban development (Chesapeake Bay Program, 2020). Like point sources, atmospheric N deposition reduction is expected to generally have limited effects on the Bay at large, since it represents only ∼17% of N input to the Bay (Ator et al., 2011). However, atmospheric N deposition can have a substantial effect on dissolved oxygen in seasons and locations where it dominates over riverine and coastal N inputs (Da et al., 2018).

4.3 | Mixed trends in N and P from nonpoint sources

Compared with the point and atmospheric sources, nonpoint sources showed mixed patterns of change during the period (Table 2 and Figures S4–S7). Only the Potomac had a statistically significant, long-term decline in TN from fertilizer. Three RIM stations, that is, Pamunkey, Mattaponi, and Choptank, had statistically significant, long-term increases in TN from fertilizer, with similar trajectories (Figure S4). However, all nine RIM stations had statistically significant, long-term declines (Figure S5) in TP inputs from fertilizer. These declines all appeared to start around 1995. For TN inputs from manure, the nine RIM stations all had statistically significant, long-term trends, with three RIM stations (i.e., Rappahannock, Pamunkey, and Patuxent) showing declines and the others showing increases (Figure S6). For TP inputs from manure, the above three watersheds also showed statistically significant, long-term declines, whereas all the other RIM stations except Potomac showed statistically significant, long-term increases (Figure S7). These diverse trends were likely driven by the intensity of row crop agriculture and the magnitude of livestock and poultry populations (Keisman et al., 2018; Sekellick et al., 2019).

For both fertilizer and manure, the below-RIM watershed contributed less TN and TP than the RIM watershed (Table 2). For fertilizer, TN from the RIM watershed had a long-term decline of −100 Mg/year ($p = 0.43$), whereas TN from the below-RIM watershed had a long-term increase of 440 Mg/year ($p < 0.01$). TP from the RIM and below-RIM watershed had a long-term decline of −640 Mg/year ($p < 0.01$) and −190 Mg/year ($p < 0.01$), respectively. For manure, TN from the RIM watershed had a long-term increase of 91 Mg/year ($p < 0.1$), whereas TN from the below-RIM watershed had a long-term decline of −120 Mg/year ($p < 0.01$). TP from the RIM watershed had a long-term increase of 28 Mg/year ($p = 0.27$), whereas TP from the below-RIM watershed had a long-term decline of −56 Mg/year ($p < 0.01$). These results indicate different progress in reducing nonpoint sources in the RIM and below-RIM watersheds: for the RIM watershed, only fertilizer TP had a statistically significant reduction; for the below-RIM watershed, fertilizer TP, manure TN, and manure TP all had statistically significant reductions.

Despite the mixed trends in nonpoint source inputs, average yields of TN from cropland to streams in carbonate settings have declined substantially (Ator et al., 2019). In addition, agricultural surplus (i.e., total input minus total output) has reportedly declined in some areas within the Bay watershed (Sabo et al., 2022). Nevertheless, continued or even strengthened management of agricultural nonpoint sources is necessary to meet the TMDL goals, which may be achieved by properly implementing best management practices (BMPs; Keisman et al., 2018; Sekellick et al., 2019). In this regard, land retirement, animal waste management systems, and conservation tillage are among the most effective BMPs for reducing N; whereas, animal waste management systems, pasture fencing, and phytase feed additives are most effective for reducing P (Sekellick et al., 2019). The potential to reduce agricultural nonpoint sources can be significant because fertilizer and manure reportedly account for 54% of N and 43% of P from the Bay watershed, respectively (Ator et al., 2011).

5 | ESTUARINE RESPONSE TO CHANGES IN NUTRIENT LOADS

In the above context of reducing major sources and riverine loads to the Bay, we present a synthesis of recent literature on estuarine response to nutrient reductions with a focus on the Chesapeake Bay water quality standards (WQS). The WQS criteria have been developed for dissolved oxygen (DO), water clarity/submersed aquatic vegetation (SAV), and chlorophyll-a (EPA, 2003; Tango & Batiuk, 2013). Considering these endpoints, the Bay TMDL is designed to reduce nutrient and sediment loads to target levels that will eventually lead to estuarine water quality that can fully support living resource survival, growth, and reproduction (EPA, 2010; Keisman & Shenk, 2013; Linker et al., 2013; Shenk & Linker, 2013). To assess the estuarine condition, a multimetric indicator was developed to integrate all three WQS criteria (Hernandez Cordero et al., 2020). This indicator showed a statistically significant increase between 1985 and 2016, which is linked to the reduction of TN from the watershed (Zhang, Murphy, et al., 2018). However, the degree of overall WQS attainment is well below 100%, stressing the need for continued nutrient reductions.

Estuarine response to nutrient reductions has also been documented for the individual WQS criteria. For DO, Zhang, Tango, et al. (2018) reported generally better DO criterion attainment in 2014–2016 than 1985–1987 for various tidal regions and salinity regimes, some of which had statistically significant, long-term improvements. In addition, Murphy et al. (2011) documented an overall decline in late-summer hypoxia in the Bay due to the reduction of N from Susquehanna River. Also, Murphy et al. (2022) reported that changes in riverine loads and below-RIM point source loads of nutrients are responsible for changes in estuarine nutrient concentrations at long-term monitoring stations throughout the Bay. Ni et al. (2020) documented modest DO increases associated with nutrient reductions despite the dominance of warming. Similarly, Frankel et al. (2022) reported DO improvements due to nutrient reductions despite increasing climate impacts. Moreover, Irby and Friedrichs (2019) projected DO improvements in the tidal waters because of nutrient reductions. Scavia et al. (2021) projected a significant reduction in hypoxic volume in the Bay associated with reductions in FN nitrogen load. Furthermore, Li et al. (2016) concluded that variability in N load is the main driver of interannual variability of hypoxia in the Bay. On a more local scale, Tian (2020) reported that nutrient loads determine the intensity and variability in hypoxia in the Chester River estuary.

For water clarity/SAV, Lefcheck et al. (2018) documented that SAV in Chesapeake Bay has achieved its highest cover in almost half a century, owing to N reductions. In addition, Ruhl and Rybicki (2010) reported increased SAV abundance and diversity in the tidal Potomac River, which are significantly linked to reductions of in situ nutrients, wastewater-treatment effluent N, and total suspended solids. For the tidal Patuxent River, Boynton et al. (2008) observed SAV reestablishment due to nutrient removal technology upgrades at wastewater treatment plants. For the Susquehanna Flats, Gurbisz and Kemp (2014) documented an unexpected resurgence of a large SAV bed and attributed water quality as an important driver. For the entire Bay, Testa et al. (2019) suggested region-specific responses of Secchi depth to in situ variables (TSS or chlorophyll-a), which have been targeted by management actions under the TMDL.

For algal biomass (chlorophyll-a), bioassay samples have shown strong spatial and seasonal variability in nutrient limitation in the Bay's mainstem (Fisher et al., 1999; Kemp et al., 2005), stressing the importance of controlling both N and P (Paerl, 2009). Zhang et al. (2021) estimated modest changes in nutrient limitation in the mainstem between 1992–2002 and 2007–2017, with expanded areas of N-limitation, suggesting the effects of N reduction. Zhang et al. (2022) also predicted that three major tributaries to Chesapeake Bay have become more limited by nutrients following nutrient reductions. Similarly, Harding et al. (2016) noted a partial reversal of nutrient over-enrichment due to modest decreases of N load since the 1980s. Harding et al. (2019) noted that the response of chlorophyll-a to N load varies by salinity zone and season and that further reductions of N load are necessary to attain chlorophyll-a criteria. On a more local scale, Boynton et al. (2013) documented that nutrient reductions from a wastewater treatment plant have resulted in a significant decline in chlorophyll-a in the tidal Mattawoman Creek, which led to improvements in SAV coverage and water clarity. However, Wainger et al. (2016) reported that the relative influence of local and regional drivers of chlorophyll-a can vary by estuarine location. Specifically, chlorophyll-a is expected to be more responsive to nutrient reductions in well-flushed locations.

6 | EMERGING CHALLENGES TO NUTRIENT AND SEDIMENT REDUCTIONS

6.1 | Susquehanna River reservoirs can no longer effectively trap sediment and phosphorus

An emerging challenge to the Bay restoration efforts is the filling of sediment in Conowingo Reservoir and two upstream reservoirs on the Susquehanna River (Linker, Batiuk, et al., 2016). After 90 years of operation, the Conowingo Reservoir has almost reached its sediment storage capacity (Langland, 2015) and entered a state called "dynamic equilibrium," while the other two reservoirs have reached that status for decades (Cerco, 2016). Historically, the reservoir could trap on average 2% of TN, 45% of TP, and 70% of SS from the upstream watershed (Langland, 2009; Langland & Hainly, 1997). However, it can no longer effectively trap sediment and sediment-attached nutrients (Hirsch, 2012; Zhang et al., 2013; Zhang, Hirsch, & Ball, 2016). Such change in trapping efficiency has occurred under both storm-flow and normal-flow conditions (Zhang, Hirsch, & Ball, 2016).

These changes in reservoir performance point to the need for greater load reductions to achieve the TMDL (EPA, 2010). This presents a significant challenge because the Susquehanna is the largest tributary to the Bay, accounting for 65% of TN, 46% of TP, and 41% of SS from the RIM watershed (Zhang et al., 2015). The Bay Program Partnership has incorporated recent scientific knowledge (Cerco, 2016; Hirsch, 2012; Langland, 2015; Linker, Batiuk, et al., 2016; Linker, Hirsch, et al., 2016; The Lower Susquehanna River Watershed Assessment Team, 2015; Zhang et al., 2013) to upgrade its watershed model to represent the temporal changes in reservoir function, which is used to inform the allocation of the additional load reductions (Chesapeake Bay Program, 2017). Moreover, a separate watershed implementation plan has been developed for Conowingo Reservoir, through which the jurisdictions will combine resources and reduce implementation costs by targeting reduction practices in the most effective areas (Center for Watershed Protection, 2021).

The sediment buildup behind Conowingo Dam has differential effects on P and N delivery. In particular, the sediments are mostly fine grained (Zhang & Blomquist, 2018), thereby providing large surface areas for transporting P. Although the loss of trapping capacity has the biggest impact on sediment delivery, the associated increase in particulate P is expected to have a larger impact on the estuary (Linker, Batiuk, et al., 2016). Furthermore, there is growing evidence that orthophosphate exiting Conowingo Dam has increased substantially since the late 2000s, which is a highly bioavailable form of P for phytoplankton (Fanelli et al., 2019). However, the filling up of Conowingo Reservoir should have limited impact on N delivery, which is mostly dissolved. Phytoplankton in the Bay is controlled by N in critical seasons and locations (Fisher et al., 1999; Kemp et al., 2005; Zhang et al., 2021). Thus, the Bay jurisdictions could continue the efforts to reduce N from major sources, particularly agricultural nonpoint sources.

6.2 | Legacy nutrients, climate change, and population growth present additional challenges

Stakeholders of the Bay restoration efforts recognize that restoration progress is dependent on turning the tide of decades of nutrient applications and storage in the watershed (Hirsch et al., 2013). For example, riverine TN has not yet shown a decline at the Choptank RIM station due to accumulation of historically applied N in the groundwater (Ator & Denver, 2015; Hirsch et al., 2010; Zhang et al., 2015). Among the RIM tributaries, time lags to achieving water quality vary considerably, with the longest lags attributed to the Choptank, where N surpluses in the watershed remain high (Chang et al., 2021). Although legacy N primarily accumulates in the groundwater (Bachman et al., 1998; Sanford & Pope, 2013), legacy P primarily accumulates in soils and river bed sediments (Ator et al., 2011, 2020; Kleinman et al., 2019; Vadas et al., 2018). Although the progress of restoration efforts can be hindered by legacy nutrients (Jarvie et al., 2013; Meals et al., 2010; Van Meter et al., 2018), more rapid progress may be achieved by targeting management practices on nutrient sources that can bypass the legacy stores, including those applied to areas associated with shorter transit times.

Climate change is another important challenge to the Bay restoration. Projected increase in air and water temperature and acceleration of the water cycle (Hinson et al., 2021; Milly et al., 2005; Najjar et al., 2010; Rice & Jastram, 2014) can alter riverine water quantity and quality and hinder the progress of load reductions. For example, Sinha et al. (2017) estimated that changes in precipitation alone will substantially increase riverine TN within the continental United States. Moreover, the effects may be more pronounced in critical seasons. For Chesapeake Bay, accelerated water cycle is expected to increase river runoff and nutrient and sediment loads during the winter–spring season (Wagena et al., 2018), which is critical for hypoxia in the Bay (Murphy et al., 2011). Climate change can further exacerbate the issue by inducing changes in agricultural management such as earlier spring tillage and altered nutrient application timing (Wagena et al., 2018). In a more recent study, however, Ator et al. (2022) predicted a slight net decline in annual N delivery to the Bay between 1995 and 2025 due to effects of expected climate change. The authors estimated that increases in N due to a wetter climate are more than offset by increasing rates of denitrification, ammonia volatilization, and changes in plant phenology (Ator et al., 2022). Sea level rise will also likely affect tidal height and coastal erosion rates (Sanford & Gao, 2018). In the above context, the Bay Program Partnership has been actively evaluating the impacts of climate change on BMP effectiveness, nutrient export, and estuarine water quality, providing critical information for the restoration effort (Frankel et al., 2022; Johnson et al., 2018; Najjar et al., 2010; Ni et al., 2020; Tian et al., 2021; Xu et al., 2019).

Population growth and associated urbanization pressures have the potential to offset some progress in reducing loads to the Bay. Human population in the Bay watershed increased by 117% between 1950 and 2017 and is projected to exceed 20 million by 2030 (Chesapeake Bay Program, 2020). This will likely lead to urban and suburban expansions, resulting in increases in impervious surface areas and stormwater runoff, wastewater effluent volume, and possibly atmospheric deposition due to augmentation in vehicle emissions. These additional sources, if not managed properly, could hinder the achievement of TMDL goals.

7 | CONCLUSION: REFLECTIONS AND INSIGHTS

This synthesis provides a clear perspective on a key management question: *After three decades of restoration, has there been progress made in reducing nitrogen, phosphorus, and sediment from the Bay watershed?* We conclude with some reflections and insights.

Nitrogen load has decreased in the RIM watershed, but phosphorus and sediment loads have lacked progress. These broad patterns are consistently shown in the FN loads and true-condition FWCs. Nitrogen reductions to date are largely driven by reductions in point sources and atmospheric deposition. Future nitrogen reductions from the RIM watershed will require significant progress in managing agricultural nonpoint sources, including fertilizer and manure. The progress in N reduction is encouraging because N is the predominant driver of hypoxia in Chesapeake Bay (Kemp et al., 2005; Murphy et al., 2011; Scavia et al., 2021) and the limiting nutrient of summer phytoplankton growth (Fisher et al., 1999; Kemp et al., 2005; Zhang et al., 2021). The N reduction has reportedly led to numerous improvements in estuarine water quality (Lefcheck et al., 2018; Murphy et al., 2011; Zhang, Murphy, et al., 2018). The lack of progress in reducing P and sediment, however, is notable in the context of the TMDL. P has ecological implications since it is the limiting nutrient of spring phytoplankton growth in low-salinity regions (Fisher et al., 1999; Kemp et al., 2005; Zhang et al., 2021), and sediment can increase light attenuation and adversely impact submerged aquatic vegetation in the upper regions of the estuary. Thus, the Bay Program Partnership could strive to refine strategies to expedite the reduction of P and sediment loads.

Although a majority of the Bay watershed is monitored at the RIM stations (78.3% by flow), the unmonitored below-RIM watershed comprises a disproportionately high fraction of nutrient and sediment inputs to the Bay. Therefore, managing nutrient inputs from the below-RIM watershed is crucial for achieving the TMDL goals. In this regard, our analysis of the major sources for the below-RIM watershed showed encouraging results that point sources (both N and P), atmospheric deposition (N), manure (N and P) and fertilizer (P) all had long-term declines, which are expected to result in water-quality improvements in the adjacent tidal waters. Understanding the resulting response of water-quality patterns and trends could require additional monitoring in the below-RIM watershed.

Nutrient trends and sources vary considerably among the RIM tributaries, creating opportunities for targeted management actions. Results show diverse trends in TN, TP, and SS FWCs among the RIM tributaries, highlighting areas that require more intensive management. Declines in TN FWC were observed for the three largest tributaries plus Patuxent, whereas declines in TP FWC were observed for the James and Patuxent. By contrast, the Pamunkey and Choptank showed increasing trends in both TN and TP FWC. The latter observation is concerning, because it may represent the much larger (but unmonitored) Eastern Shore (Ator & Denver, 2015). Results also show diverse trends in the major sources, that is, wastewater and atmospheric deposition appear to be the primary drivers of nutrient reductions, with generally limited progress in nonpoint sources. In addition, N and P differ in terms of transport and transformation (Ator et al., 2011, 2019, 2020), which could be factored into management by developing strategies that include the necessary mixture of practices to target the delivery of both nutrients. Another consideration is nutrient speciation, that is, N is mainly dissolved, whereas P is mainly particulate (Zhang et al., 2015). The dissolved fractions of N and P could be given high management priority because of their potency to fuel phytoplankton growth (Shenk et al., 2020).

Spatially and temporally targeted restoration actions should lead to disproportionately large benefits in improving water quality. Chesapeake Bay and its watershed are a complex ecosystem, which is reflected in many facets, including the spatial and temporal variability in nutrient inputs, the lag times in groundwater and soils, the water-quality patterns in the rivers and the Bay, the residence time and mixing in the tidal waters, and the underlying controls (physical vs. biogeochemical; Ator et al., 2020; Ator et al., 2019; Kemp et al., 2005; Noe et al., 2020; Testa et al., 2019; Wainger et al., 2016). Consequently, different locations of the Bay can exhibit divergent responses and restoration may be most effective if local-specific characteristics are considered (Wang et al., 2015). Moreover, not all nutrients entering the Bay are equal in terms of their impacts on estuarine water quality (Fisher et al., 1999; Kemp et al., 2005). Similarly, not all nutrients applied on the land are equal in terms of their impacts on the riverine water quality (Ator et al., 2011). In fact, the greatest yields of N delivered to the Bay are contributed from agricultural areas of the northern Piedmont and Valley and Ridge (Ator et al., 2011). Such spatial inequality of influence points to opportunities for the Partnership to target reduction efforts on critical locations to achieve disproportionately large benefits in water-quality improvement. Similarly, there is temporal inequality in nutrient export, and restoration efforts could consider targeting reduction efforts on critical events to achieve cost effectiveness (Preisendanz et al., 2020).

There has been measurable progress in nutrient reductions over the last three decades, but these measures fall short of goals and expectations. Our synthesis of literature and published data demonstrate that the Bay cleanup efforts have achieved some progress in the last three decades and that nutrient reductions appear to be effective in improving Chesapeake Bay with respect to the key water quality standards, that is, DO, chlorophyll-a, and water clarity/SAV. Although signals of ecosystem recoveries are observed, Chesapeake Bay has not yet been fully restored. Continued reductions of nutrients and sediment under the Bay TMDL are critical to ensure that successes achieved to date will not languish or be reversed (Zhang, Murphy, et al., 2018). In this context, challenges due to Conowingo Reservoir, legacy nutrients, climate change, and population growth should continue to be considered by the Bay Program Partnership (Hirsch et al., 2013; Johnson et al., 2018; Linker, Batiuk, et al., 2016).

Continued monitoring, modeling, and assessment are critically important for informing adaptive management of this ecosystem. This is demonstrated by the published data and research over the last three decades, which made this synthesis possible. Moving forward, it should be noted that ecosystems such as Chesapeake Bay may never return to their reference status even if the human pressures are reversed (Duarte et al., 2009). In this regard, point source dominated ecosystems tend to improve more rapidly and linearly in response to nutrient reductions. However, in nonpoint source dominated ecosystems such as Chesapeake Bay, responses tend to be more non-linear with hysteresis and time lags (Kemp et al., 2009). Therefore, continued monitoring, modeling, and assessment are important for measuring progress, capturing recovery trajectory, and understanding the underlying mechanisms. Past and future advancements in the scientific understanding of Chesapeake Bay and its watershed are valuable resources that can inform the restoration of other ecosystems (Boesch, 2019; Kemp et al., 2005).

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The authors have declared no conflicts of interest for this article.

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The data that support the findings of this study are available in the public domain: https://doi.org/10.5066/P96NUK3Q and https://cast.chesapeakebay.net/.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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